8. River Transport Module

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The Columbia River represents the final link in the environmental pathway through which contaminants reach various receptors such as humans, plants, or animals.

The River Transport Module simulates the Columbia River between the Vernita Bridge and McNary Dam including inputs from groundwater, the Yakima River and the Snake River (Figure 8.1).

The contaminants modeled in the river come from three sources:

- Those already in the river when water reaches the Vernita Bridge from upstream sources and atmospheric fallout.
- Contaminant influx from Hanford waste sites through groundwater.
- Direct discharge to the river from Hanford facilities.
- Groundwater and irrigation return discharges to the river along the shore opposite Hanford are not included in the initial assessment.

The Modular Aquatic Simulation System 2D (MASS2) code provides the basis of the River Transport Module (Richmond et al. 2000). MASS2 is a two-dimensional, depth-averaged hydrodynamics model that provides the capability to simulate the lateral (bank-to-bank) variation of flow and transport of sediments and contaminants. The model incorporates river hydraulics (velocity and water depth), contaminant influx to the river through groundwater and point sources, sediment and contaminant transport, and adsorption/desorption of contaminant to sediments.

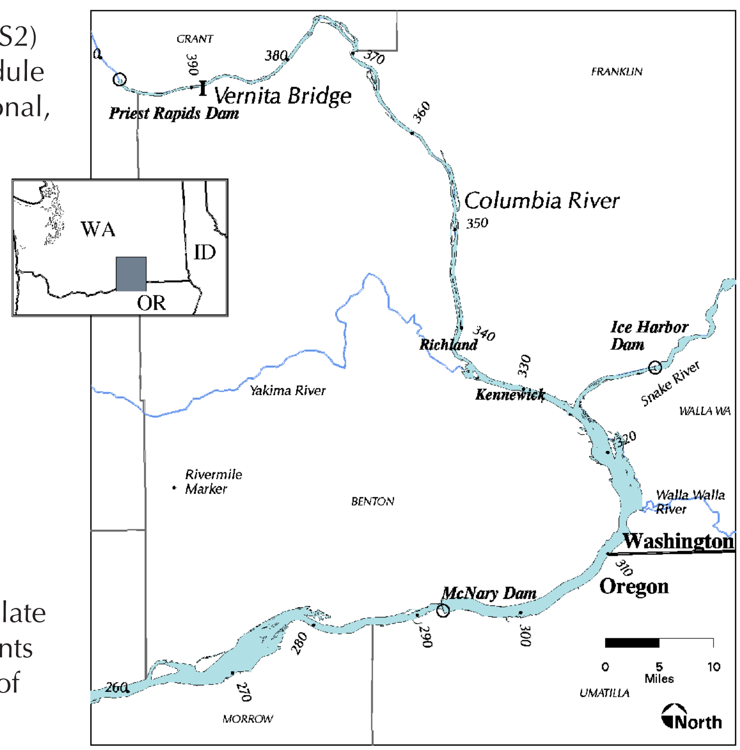

__Figure 8.1.__ Columbia River showing the area between Vernita Bridge and McNary Dam included in the River Module.

Results

MASS2 was run for a set of 25 realizations to simulate the transport and fate of 10 radioactive contaminants in the Columbia River. Simulation results for two of these contaminants, tritium and uranium, are discussed in this chapter.

A plot of the river near the 300 Area shows the tritium plume concentrated along the Hanford

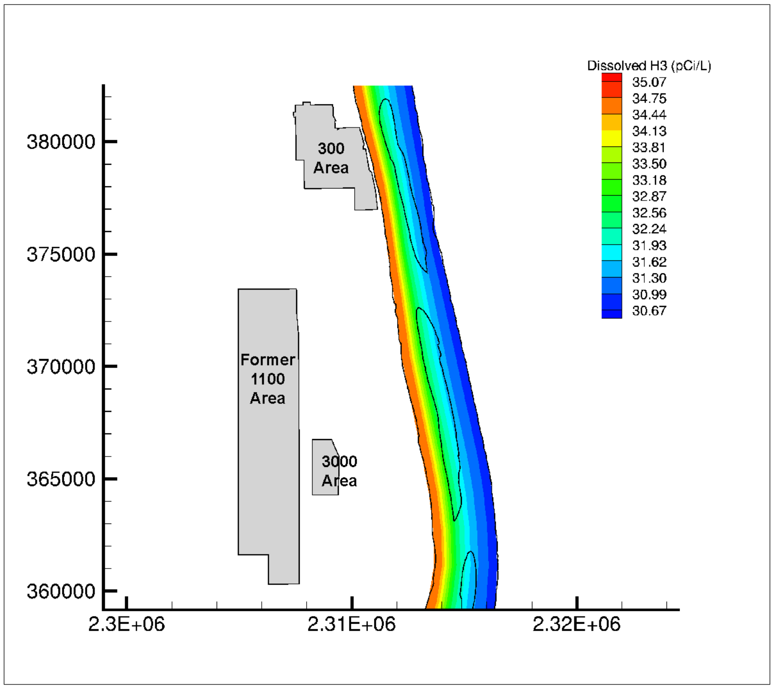

Figure 8.2. *Modeled dissolved tritium concentrations in the Columbia River near the 300 Area in 1995.*

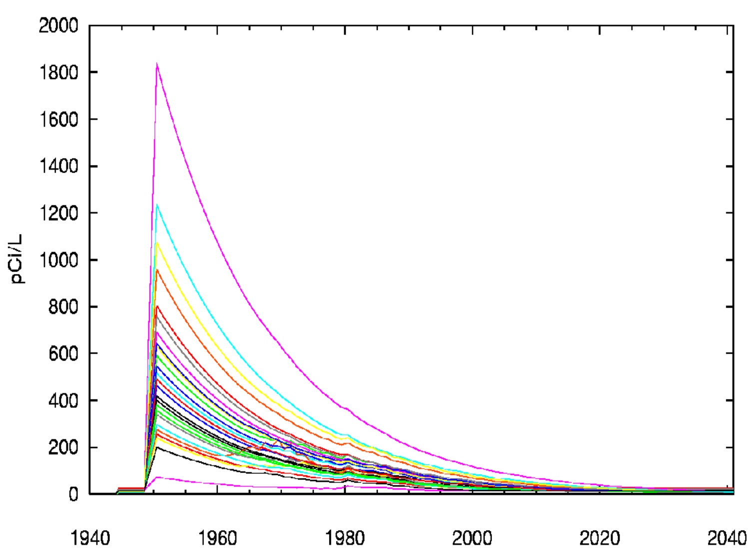

Figure 8.3. *Time series plot of dissolved tritium in surface water at the Richland Pump House for all 25 realizations.*

(western) shore of the river (Figure 8.2). The higher concentrations along the Hanford shore are caused by tritium influx from groundwater. This plume extends from upstream, near the 100 Areas, downstream to well below the mouth of the Yakima River. (Note: the islands shown in the river shoreline layer are not included in the computational grid.)

Tritium concentrations at the Richland Pumphouse appear to be largely dominated by inflow concentrations at Vernita resulting from global atmospheric background levels associated with fallout from nuclear weapons testing done in the 1950s and early 1960s (Figures 8.3 and 8.4). The fallout is represented by a concentration spike occurring around 1950. The tritium concentration corresponding to atmospheric nuclear weapons testing declines due to radioactive decay. The effects of tritium inputs into the river from groundwater on concentrations at the Richland Pumphouse are evident where fluctuations above background occur from as early as 1970 until the 1980s. Modeled results for the year 2000 indicate that the tritium from groundwater may cause a 5 to 35% increase in concentrations in the river. By comparison, the concentrations in the river are less than 0.5% of the drinking water standard. Tritium in the groundwater reaches the river relatively quickly compared to the other contaminants because it does not adsorb to sediment. Therefore, it is very mobile and passes through the groundwater system essentially as a water molecule.

The effect of uranium-238 from Hanford is more difficult to see in the river. The input of Hanford-related uranium-238 to the river is small compared with naturally occurring background uranium, and the greater tendency of uranium to adsorb to sediment. The effect of Hanford-related inputs on the dissolved uranium-238 concentration in the surface water appears to be negligible relative to the background concentration (Figure 8.5) in that Hanford-derived uranium-238 contributes less than 0.5% of the background value for all realizations. The differences in concentrations from the Richland Pump House and McNary Dam locations are also negligible. The particulate uranium-238 concentrations in McNary Dam surface water are higher, however (see Figure 8.5, bottom). This is probably due to the additional time available for dissolved uranium-238 to adsorb to suspended sediments while enroute to McNary Dam and inputs from the Yakima and Snake Rivers.

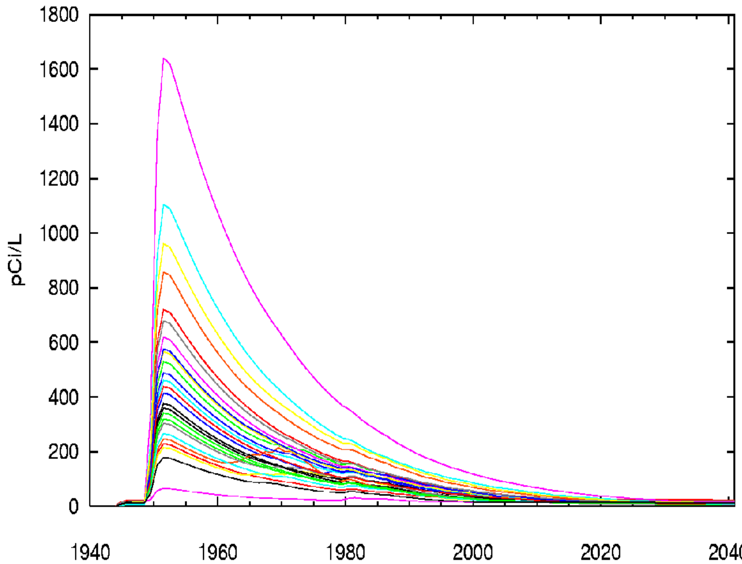

Figure 8.4. *Time series of dissolved tritium in the riverbed pore water at Richland Pump House for all 25 realizations.*

Conceptual Model

The Columbia River is the largest North American River to discharge into the Pacific Ocean. The river originates in Canada and flows south 1,953 kilometers (1,212 miles) to the Pacific Ocean. The watershed drains a total of 670,000 square kilometers (258,620 square miles) and receives waters from seven states and one Canadian province. Key contributors to the flow are runoff from the Cascade Mountains in Washington and Oregon and from the western slopes of the Rocky Mountains in Idaho, Montana, and British Columbia. Average annual flows below Priest Rapids and The Dalles dams are approximately 3,360 cubic meters (120,000 cubic feet) per second and 5,376 cubic meters (192,000 cubic feet) per second, respectively. Numerous dams within the United States and Canada regulate flow on the main stem of the Columbia River. Priest Rapids Dam is the nearest dam upstream of the Hanford Site, and McNary Dam is the nearest downstream. The dams on the lower Columbia River greatly increase the water travel times from the upper reaches of the river to the mouth, subsequently reducing the sediment loads discharged downstream. The increased travel times also allow for greater radionuclide deposition and decay.

The conceptual model includes data about the groundwater/river interface and complex river dynamics to evaluate how contaminants move through the river system.

The Snake, Yakima, and Walla Walla Rivers all contribute suspended sediment to the Columbia River; contributions from the Snake River are the most significant. Since construction of McNary Dam (completed in 1953), much of the sediment load has been trapped behind the dam. However, at

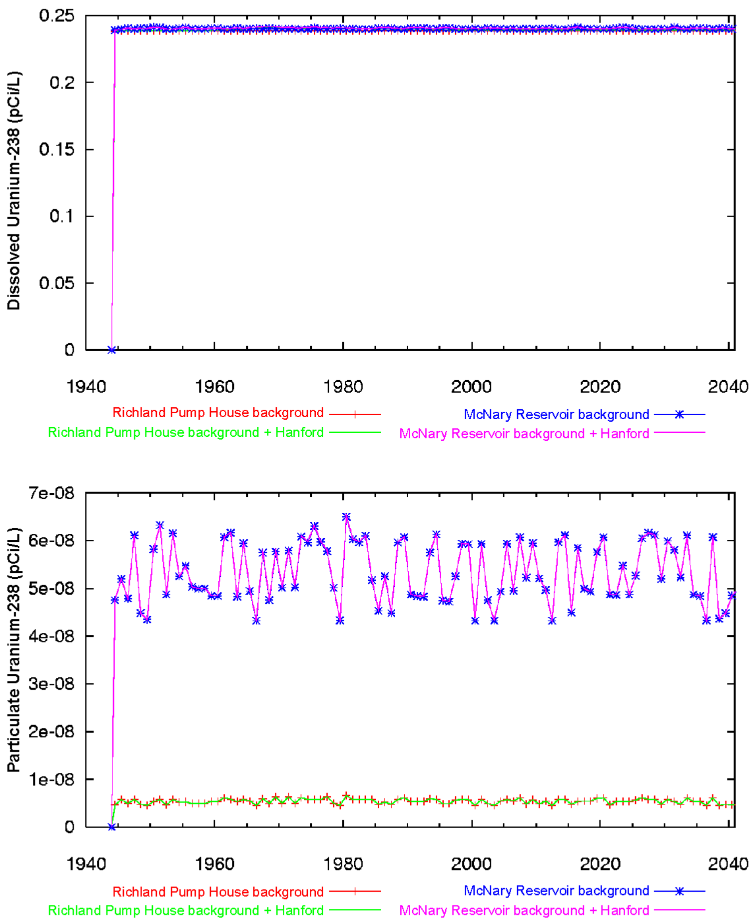

Figure 8.5. *Time series plots of dissolved (upper) and particulate (lower) uranium-238 in the surface water column at the Richland Pump House and McNary Reservoir. The background only (bg) and bg + Hanford results are plotted together; however, the Hanford input does not result in a noticeable increase in uranium concentration. Note that the initial condition is set to zero in the model, and the concentrations immediately jump to the background value.*

McNary Dam and other Columbia River dams, some of the trapped sediment is resuspended and transported downstream by seasonal high discharges. As expected, much of this material is redeposited behind dams located farther downstream. Sediment accumulates faster on the Oregon shore than on the Washington shore because sediment input from the Snake and Walla Walla Rivers stay near the shore on the Oregon side. Sediment-monitoring samples taken for the Hanford Sitewide Surface Environmental Surveillance Project indicated cobble and coarse- and fine-sand bed sediments at sampling locations along the Hanford Site (Blanton et al. 1995). Silt and clay sediment was observed at the McNary Dam sampling site.

The conceptual model used in the initial assessment included the environmental pathways and transport processes that affect contaminant transport in surface water systems. These pathways and processes are illustrated in Figure 8.6. The key assumption in the conceptual model is that it includes all important features, processes, and events related to the fate and transport of contaminants in the Columbia River system. The initial assessment River Module included these processes in the mathematical implementation of the conceptual model.

Several sources cause uncertainty in the mathematical representation of the conceptual model. These include the choice of temporal and spatial scales, initial and boundary conditions, model parameters, and the physical processes themselves. Examples of uncertainty in physical processes are fluid turbulence and cohesive sediment transport. Uncertainty also arises when selecting parameters such as channel roughness coefficients, porosity, and sediment-contaminant interaction coefficients as well as the influx of contaminants through the interface with groundwater. Uncertainty in the initial assessment was addressed primarily using Monte

Carlo simulations and sensitivity studies. Uncertainty results are discussed in Chapter 11.

Implementation Model

The input/output processes implemented include influxes from incoming flow and tributaries, groundwater and the vadose zone, and removal by outflow. The physical processes include river hydrodynamics and suspended sediment transport, deposition, and resuspension. Because of run-time constraints, suspended and bed sediments were modeled with only the silt-size fraction. Finally, the contaminant transport processes include surface water advection and dispersion, sorption/desorption to sediments, decay, and exchange between bed pore water and the overlying surface water. In addition, a coarse grid was developed to decrease overall computational time and allow timely simulation of the full 1,000-year time frame.

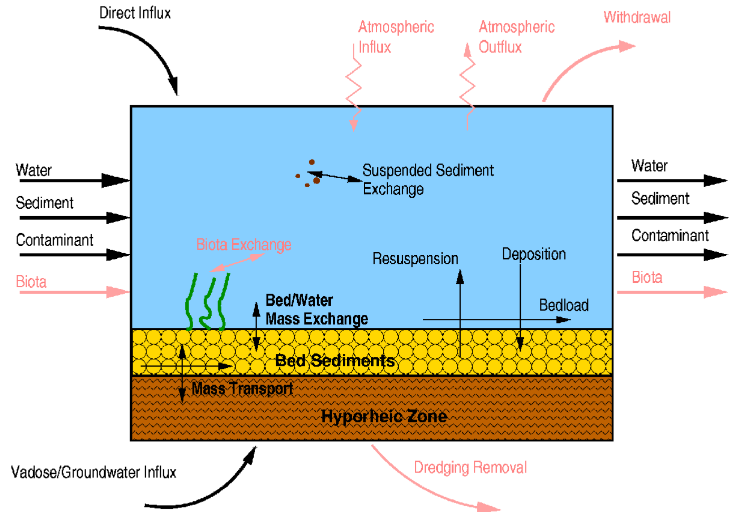

Figure 8.6. *Schematic of the transport and fate processes in the river conceptual model. Processes included in the implementation model are shown in black. Processes reserved for future implementation appear light pink.*

Numerical Model

The River Flow and Transport Model, MASS2 (Richmond et al. 2000), includes the capability to simulate sediment transport, sediment-contaminant partitioning (using K_ds), sediment-sorbed contaminant transport, and contaminant transport within the riverbed sediment layer.

The model is an unsteady finite-volume code formulated using the general principles described by Patankar (1980). The model uses a structured multiblock scheme on an orthogonal curvilinear grid system. The momentum and mass conservation (continuity) equations are coupled using a variation of Patankar's (1980) SIMPLE algorithm extended to shallow-water flows by Zhou (1995). Spasojevic and Holly (1990) give an example of a two-dimensional model of this type. The governing equations are formulated in a conservation form using a full-transformation in the curvilinear system (Richmond et al. 1986). The governing equation for the transport of a contaminant is obtained by applying the principle of conservation of mass to a fluid element. The model is coded in standard FORTRAN 95 and runs on several operating systems including Windows, Unix, and Linux.

The conceptual model for the River Module takes into consideration a large number of different biological, chemical, and physical processes that control the transport of contaminants to the river ecosystem.

To numerically solve the system of governing equations, initial and boundary conditions must be specified. Initial conditions for each dependent variable (velocity, depth, and species) are assigned at the start of each simulation either as approximate values or using the results of a previous simulation (i.e., hotstart or restart file). At the upstream boundary, the incoming velocity or discharge is specified as a function of time for each cell, and depth is extrapolated from the nearest interior cell. At the downstream boundary, the depth for each cell is specified as a function of time, and zero-gradient conditions are assigned for the velocity. Along the shoreline, a zero gradient or slip condition is applied to the longitudinal velocity component, and the normal velocity to the shore is set to zero. The depth is extrapolated from the nearest interior cell to the shore.

River Inflows and Stages. Data on river discharges were obtained from U.S. Geological Survey gage data. The Vernita Bridge, Kiona, and Burbank gages were used for the Columbia, Yakima, and Snake River flows, respectively. Because of a gap in the Snake River data, some project operations data at Ice Harbor Dam were used in addition to the gage data. The water surface elevation of the McNary forebay was held constant at the normal operating stage of 104 meters (340 feet) above mean sea level for all runs.

Table 8.1. *Summary of K_d values (in L/g or m^3/kg) used for suspended sediment in the river for the initial assessment.*

Contaminant	Minimum	Medium	Maximum
Carbon Tetrachloride	0.0002	0.0601	0.12
Chromium	0.006	1.5	2.5
Tritium	0	0	0
Strontium-90	0.0036	16.002	32
Technetium-99	0.001	0.1705	0.34
Iodine-129	0.0002	0.1851	0.37
Cesium-137	0.037	15.769	31.5
Uranium	0.001	3.0005	6
Plutonium-239	0.027	159.51	319

Distribution Coefficient (K_d) Values. Distribution coefficient (K_d) estimates for the linear sorption isotherm model are site-specific because they are affected by numerous site-specific characteristics including pH, salinity, substrate size and composition, substrate cation exchange capacity, the presence of organics, the concentrations of competing ions, and redox potential. The K_d values shown in Table 8.1. were derived from directly applicable local studies and other studies involving fresh water aquatic and groundwater environments with basalt substrates.

Table 8.2. *Suspended sediment concentrations (kg/m^3) used in the river model.*

Location	Concentrations (kg/m^3)
Vernita Bridge	0.00375
Yakima River	0.06
Snake River	0.016

Suspended Sediment. The background suspended sediment data were obtained from the U.S. Geological Survey National Stream Water Quality Network Web site, http://water.usgs.gov/nawqa. Data for the Columbia River at Vernita Bridge, the Yakima River at Kiona, and the Snake River at Burbank were downloaded from the Web site and used as model boundary conditions. All of the suspended sediment concentration data were averaged for each location to estimate the background suspended sediment concentration. The data shown in Table 8.2, consisted of several measurements per year starting in 1996.

Radionuclide Data. The initial assessment used upriver background values to establish baseline concentrations in the Columbia River. The background values used for the initial assessment were based on 1990 to 1995 surface water data from the Columbia River Comprehensive Impact Assessment, the Hanford Environmental Information System (HEIS) for the years 1996 to 1999, and data from the U.S. Geological Survey measured specifically in the confluence areas. The Columbia River (Vernita Bridge) background values were obtained from the Columbia River Comprehensive Impact Assessment Segment 1 data. Only limited data were available for the Snake and Yakima Rivers. When no sampling data were available in these areas, the data distributions from the Columbia River at Priest Rapids Dam were used. Background concentrations for contaminants sorbed to sediment were calculated from the surface water concentrations assuming equilibrium partitioning between the dissolved fraction and the fraction sorbed to sediment. Background concentrations for the river-bottom pore water were set equal to the surface water concentrations.

Background concentrations based on sampled data were used for the Snake River, Yakima River, and Columbia River upstream of the Vernita Bridge. These concentrations formed the backdrop on which Hanford releases were superimposed.

Table 8.3. *Background surface water data values for the initial assessment.*

Contaminant	Location	Geometric Mean	Units	Geometric Standard Deviation	Source	Basis
Cr-VI	Snake River	0.701	µg/L	1.77	New Calculation, USGS[b] Data	10-year geometric mean for 1990-2000
Sr-90	Snake River	1.09	pCi/L	2.09	New Calculation, USGS Data	10-year geometric mean for 1990-2000
Cr-VI	Yakima River	0.478	µg/L	1.10	New Calculation, USGS Data	10-year geometric mean for 1990-2000
H-3	Columbia River	39.7	pCi/L	1.81	CRCIA Project[a]	5-year geometric mean for 1990-1995
Cs-137	Columbia River	8.00E-04	pCi/L	29.9	CRCIA Project	5-year geometric mean for 1990-1995
Sr-90	Columbia River	0.0854	pCi/L	1.86	CRCIA Project	5-year geometric mean for 1990-1995
Tc-99	Columbia River	0.0299	pCi/L	10.3	CRCIA Project	5-year geometric mean for 1990-1995
I-129	Columbia River	2.14E-08	pCi/L	56.8	CRCIA Project	5-year geometric mean for 1990-1995
CCl_4	Columbia River	0.147	µg/L	1.25	New Calculation	5-year geometric mean for 1995-2000
Cr-VI	Columbia River	0.871	µg/L	2.55	CRCIA Project	5-year geometric mean for 1990-1995
Pu-239	Columbia River	2.74E-05	pCi/L	2.83	New Calculation	5-year geometric mean for 1995-2000
U-238	Columbia River	0.173	pCi/L	1.92	CRCIA Project	5-year geometric mean for 1990-1995

(a) Columbia River Comprehensive Impact Assessment Segment 1 data.
(b) U.S. Geological Survey.

A geometric mean and geometric standard deviation were computed for each contaminant and location for which data were available. These values were used to set parameters for the stochastic distribution for the 25 realizations. The values used for the surface water background are shown in Table 8.3. Some contaminants (tritium, strontium-90, technetium-99, iodine-129, and cesium-137) are fallout products from atmospheric nuclear weapons testing. Pre-1995 data for these contaminants were obtained by back-calculating radioactive decay. The reference data year for back-calculating decay was 1995, and fallout introduction to the watershed was assumed to occur in 1950.

History Matching

MASS2 results were compared to historical data to verify both the physical hydraulic and transport model components. The simulated contaminant concentrations that MASS2 generated were compared to two sets of measured data: (1) chromium-51 and zinc-65 from 1964 to 1966, when these radionuclides were directly discharged to the river from once-through cooled plutonium production reactors, and (2) tritium and uranium from 1992 to 1996, when contaminants entered the river from groundwater sources and upstream inputs. To verify the river hydraulics component of MASS2, model results generated using both the initial assessment and a more finely resolved grid were compared to acoustic doppler current profiler measurements of water velocities made in six locations in the Hanford Reach each during September 2000 (Guensch and Richmond 2001).

River model results are compared to historical data to verify both the physical, hydraulic, and transport model components.

Hydraulics. Simulated river velocities were compared to acoustic doppler current profiler data. These data were collected in September 2000 at the following locations within the Hanford Reach: Vernita Bridge, 100 N Area, 100 F Area, Hanford townsite, 300 Area, and the Richland Pump House. At each location, two transects were taken laterally across the river. Each transect provided vertical profiles of three-dimensional velocity vectors and the depth every few feet across the channel. The vertical average of the velocity vectors was computed at each point and compared to the simulated velocities.

The simulated velocities generated using the initial assessment grid were interpolated to the acoustic doppler current profiler data points using linear regression. The two sets of velocity vectors were overlain to visually compare vector magnitudes and directions. Generally, the simulated velocities were accurate. The exceptions occurred in locations where islands were absent from the simplified grid. At these locations, the model under estimated velocities because without the islands the cross-sectional flow area is larger.

The simulated velocities generated using more finely resolved grid generally correspond well to the measured data, although the model does not reproduce all the small-scale variations in the measured cross-sectional velocity profile. This is most likely because the resolution of the grid and underlying bathymetric data do not capture the small-scale features causing the velocity irregularities. New shoreline and bathymetry surveys would help to improve hydraulic simulations.

The results from the River Transport Module appear reasonable and any discrepancies most likely result from the uncertainty of input data.

Radionuclide Transport. Monitoring data on radionuclide concentrations in the river were used to assess the performance of the fate and transport component of the model by comparing results to those of the history matching runs. Data on downstream concentrations of chromium-51 and zinc-65 at the 300 Area, Richland Pumphouse, Pasco, and the McNary Dam forebay were obtained from Walters et al. (1994). The simulated tritium and uranium-238 concentrations were compared to monitoring data from the 300 Area and Richland Pumphouse. The results of these comparisons are presented graphically and assessed statistically in the description of Columbia River history matching results in Guensch and Richmond (2001).

The primary parameters that could be adjusted to refine the contaminant transport and adsorption results were the lateral diffusion coefficient (k_y), and the distribution coefficient (K_d). The k_y value accounts for both the diffusive and dispersive mechanisms of lateral transport. Values of k_y between 0.46 and 0.93 square meters (5 and 10 square feet) per second provided the best results. The mass transfer rate for contaminant adsorption to and desorption from suspended sediment, was generally set between 1/24 and 1/48 (hour $^{-1}$). This equilibrium rate constant value can affect model stability if set too high. The rate constant estimates on the low end of the range provided the best correspondence to measured data for each of the contaminants. The parameter values were evaluated using the mean absolute error of the predicted river concentrations compared to the measured data.

Pre-1996 monitoring data on suspended sediment would benefit restoration-era transport simulations. Measurements of site-specific K_d values for the suspended and bed sediments in the aquatic environment would dispel the considerable uncertainty associated with K_d values. Surface water samples that are filtered and analyzed for suspended sediment concentration and particulate and dissolved radionuclide concentration would be useful to verify simulation results and estimate K_d values. The background-dissolved concentration predicted for uranium-238 is slightly low compared to monitoring data. A re-assessment of the background values should be performed.

The particulate concentrations are over three orders of magnitude less than the dissolved concentrations and subsequently have very little influence on the correspondence of simulated and monitored total concentrations. The correspondence of simulated concentrations to monitoring data is reasonable, and the discrepancies most likely result from the input data.